7. BASELINE RISK ASSESSMENT METHODOLOGY

The Baseline Risk Assessment (BRA) developed for WAG 4 (DOE-ID 1999a) evaluated the risk potential associated with contaminated media at CFA. The evaluation simulated a No Action alternative, meaning that mitigative measures to reduce risk were not considered. Methodologies implemented to evaluate the baseline human health and ecological risks are outlined below, followed by a summary of the results. Three sites were found to pose unacceptable risk to human health and the environment. For those three sites, components of the risks assessment specific to the selected remedies, such as contaminants of concern, contaminant concentrations, and risk estimates, are presented in detail in Section 8.

7.1 Human Health Risk Evaluation Summary

The human health risk assessment approach used in the BRA was based on the EPA *Risk Assessment Guidance for Superfund* (EPA 1989, 1992a), INEEL Track 2 Guidance (DOE-ID 1994), and the INEEL cumulative risk assessment guidance protocol (INEEL 1995b). The tasks associated with development of the human health risk assessment included the following:

- Data evaluation

- Exposure assessment

- Toxicity assessment

- Risk characterization

- Qualitative uncertainty analysis.

These tasks are described in the subsections below.

7.1.1 Data Evaluation

Data evaluation tasks that were completed as part of the BRA included site screening, contaminant screening, and development of data sets for use in the risk assessment. The screening processes were designed to be conservative so that only sites and contaminants that clearly do not pose an unacceptable risk to human health and the environment are eliminated.

Initial site screening consisted of a review of previous risk assessments conducted for WAG 4 sites identified in the FFA/CO. As a result of the site screening, 19 of the individual sites, including the sites identified in the FFA/CO, were retained for quantitative risk assessment in the comprehensive BRA. The remaining sites either exhibited no risk potential (e.g., the site had no source of contamination) or a risk potential sufficiently below threshold values to preclude a significant contribution to cumulative risk. Individual sites with risk estimates greater than or equal to 1E-06 or hazard indices greater than or equal to 1 were retained.

Site screening also involved a CFA Facilities Analysis that evaluated all operating, abandoned and demolished non-CERCLA facilities proximal or co-located to WAG 4 CERCLA sites. The analysis assessed their potential impacts to cumulative risk estimates to ensure that all historical releases were identified and assessed. The analysis included a review of past and present operational activities at CFA, existing facilities and structures, and management control procedures for mitigating the effects of future

environmental releases of contaminants. No facilities or structures were identified in the facilities analysis that would affect the cumulative risk calculations at WAG 4.

Contaminant screening consisted of comparing maximum detected concentrations to INEEL background concentrations (INEEL 1996a) and EPA risk-based concentrations (RBCs) (EPA 1995, 1997a). The Risk Based Concentrations (RBC) used to screen contaminants were calculated using the soil ingestion, soil inhalation, and external exposure pathways for a calculated lifetime cancer risk of 1E-06 or a Hazard Quotient (HQ) of 1. The most restrictive RBC was compared to the maximum detected soil concentration of each contaminant of concern. Those contaminants that exceeded the screening criteria were identified as contaminants of potential concern and retained for quantitative analysis in the BRA. Potential exposure routes were also identified in conjunction with contaminant screening using the conceptual site models (Section 5.6).

All sampling data collected at WAG 4 sites were evaluated to determine whether the data were appropriate and adequate for use in the BRA. This evaluation was conducted generally in accordance with EPA guidance (EPA 1992a). As a result of the screening process, 19 of the individual sites including the sites identified in the FFA/CO, were retained for quantitative risk assessment in the BRA.

7.1.2 Exposure Assessment

The exposure assessment quantifies the receptor intake of contaminants of potential concern for those exposure pathways that may cause adverse effects. The assessment consists of estimating the magnitude, frequency, duration, and exposure route of contaminants to receptors. The following parameters are considered in estimating exposure assessment:

- Exposed populations

- Complete exposure pathways

- Contaminant concentrations at the points of exposure for the complete exposure pathways

- Intake rates

- Intake factors.

Both populations and exposure pathways evaluated in the WAG 4 comprehensive human health BRA are illustrated in the conceptual site models (Figure 5-2 through 5-4). Land-use assumptions and projections discussed in Section 6 were used to identify exposure scenarios, pathways, and routes.

- Exposure scenarios

  - Occupational

  - Residential intrusion

- Exposure pathways

  - Groundwater pathway (cumulative)

  - Air pathway (cumulative)

  - Soil pathway

- Exposure routes

    - Soil ingestion

    - Inhalation of fugitive dust

    - Inhalation of volatiles

    - External radiation exposure

    - Dermal absorption from soil (organics and arsenic only)

    - Groundwater ingestion (residential scenario only)

    - Ingestion of homegrown produce (residential scenario only)

    - Dermal absorption of contaminants in groundwater (residential scenario only)

    - Inhalation of volatiles from indoor use of groundwater (residential scenario only).

Contaminant concentrations at the points of exposure for complete exposure pathways were calculated using upper confidence limits (UCLs) derived from analytical data. If sufficient data were not available for calculating UCL concentrations, the maximum detected concentration was used. For radioactive contaminants, radioactive decay was incorporated into the intake calculations. No degradation mechanisms for reducing the concentrations of organic or inorganic contaminants over time were considered.

Groundwater fate and transport modeling was used to predict the maximum contaminant concentrations that could occur in the aquifer from leaching and transport of nonradionuclide and radionuclide contaminants at WAG 4. The GWSCREEN model was used to simulate the potential release of contaminants from the release sites and the transport of the contaminants through the vadose zone to the aquifer.

To calculate intake rates, default intake factors from the EPA guidance (EPA 1989, 1991, and 1992a) and Track 2 guidance for the INEEL (DOE-ID 1994) were used. In conjunction with conversion factors and site-specific contaminant concentrations, these values were used to calculate contaminant intakes. The specific exposure parameters used for each receptor and exposure pathway are given in the OU 4-13 RI/FS (DOE-ID 1999a). Generally, occupational scenarios reflect workers exposed to contaminants for 8 hours/day, 250 days/year for 25 years and residential scenarios reflect exposures to contaminants for 24 hours/day, 350 days/year, for 30 years. Standard values were used to simulate the human body (e.g., mass, skin area, inhalation rates, and soil ingestion rates).

To satisfy the objective of the comprehensive risk assessment, risks produced through the air and groundwater exposure pathways were analyzed cumulatively. Cumulative risks were estimated by calculating one risk number for each contaminant of potential concern (COPC) in air and groundwater exposure routes (e.g., inhalation of fugitive dust and ingestion of groundwater) for each collection of sites in close proximity to one another. Analyzing risks for the air and groundwater pathways in a cumulative manner was necessary because contamination from all sites within an area can contribute to local air and groundwater contaminant concentrations. Conversely, individual sites within a WAG are typically isolated from one another relative to the soil pathway exposure routes (e.g., external exposure and

ingestion of soil). As a result, site-specific soil pathway exposures were analyzed. However, the BRA is comprehensive because it evaluates risks from all known sites within WAG 4, and it is cumulative because risks from multiple sites are evaluated in the air and groundwater exposure pathways.

7.1.3   Toxicity Assessment

The toxicity assessment evaluated the relationship between intake of a substance and incidence of an adverse health effect in the exposed population. Toxicity assessments evaluate the results from studies with laboratory animals or from human epidemiological studies. These evaluations were used to extrapolate from high levels of exposure, for which adverse effects are known to occur, to low levels of environmental exposures, for which effects could be postulated. Results of these extrapolations were used to establish quantitative indicators of toxicity.

Health risks from all routes of exposure were characterized by combining the chemical intake information with numerical indicators of toxicity (i.e., slope factors for carcinogens and reference doses for noncarcinogens). Toxicity constants used in the BRA were obtained from several sources. The primary source of information is the EPA online Integrated Risk Information System (IRIS) (EPA 1997b). The IRIS database contains only those toxicity constants that have been verified by EPA work groups. The IRIS database is updated monthly and supersedes all other sources of toxicity information. If the necessary data are not available in IRIS, EPA Health Effects Assessment Summary Tables (EPA 1994a) are used. The toxicity constant tables are published annually and updated approximately twice per year. The Health Effects Assessment Summary Tables contain a comprehensive listing of provisional risk assessment information that has been reviewed and accepted by individual EPA program offices, but has not had enough review to be recognized as high-quality, agency-wide information (EPA 1994a). Toxicity profiles for the contaminant of concern (COC) addressed in the selected remedies to mitigate unacceptable risk are presented below.

7.1.3.1   Lead. Lead is classified as a metal. No critical effects due to exposures to lead have been reported. However, many organs and systems are adversely affected by lead exposure. The major target organs and systems are the central nervous system, the peripheral nerves, the kidney, the gastrointestinal system, and the blood system (Sittig 1985). Anemia can be an early manifestation of lead poisoning. Other early effects of lead poisoning can include decreased physical fitness, fatigue, sleep disturbance, headache, aching bones and muscles, digestive symptoms, abdominal pains, and decreased appetite. The major central nervous system effects can include dullness, irritability, headaches, muscular tremors, inability to coordinate voluntary muscles, and loss of memory. The most sensitive effect for adults in the general population may be hypertension (Amdur, Doull, and Klaassen 1991).

Ingestion and inhalation of lead have the same effects on the human body. Large amounts of lead can result in severe convulsions, coma, delirium, and possibly death. A high incidence of residual damage, similar to that following infections or traumatic damage or injury, has been observed from sustained exposure to lead. Most of the body burden of lead can be in the bone (ATSDR 1990). Lead effects in the peripheral nervous system are primarily manifested by weakness of the exterior muscles and sensory disturbances. Lead also has been shown to adversely affect sperm and damage other parts of the male reproductive system (ATSDR 1990). Dermal absorption of inorganic lead compounds was reported to be much less significant than absorption by inhalation or oral routes of exposure (ATSDR 1990).

Behavioral effects of lead exposure are a major concern, particularly in children. Exposure to lead can cause damage to the central nervous system, mental retardation, and hearing impairment in children. Levels of exposure that may have little or no effect on adults can produce important biochemical alterations in growing children that may be expressed as altered neuropsychological behavior (Martin 1991).

Though the ability of lead to cause cancer in humans has not been shown, EPA has classified lead as a probable human carcinogen through both the ingestion and inhalation routes of exposure. Lead classification was based on the available evidence of cancer from animal studies. Rats ingesting lead demonstrated statistically increased incidence of kidney tumors (ATSDR 1990). According to some epidemiological studies, lead workers have an increased incidence of cancer. Data used in these studies are considered inadequate to demonstrate or refute the carcinogenicity of lead to humans. The EPA has not established toxicity values for lead.

7.1.3.2 *Cesium-137.*

The radioactive isotope Cesium-137 is a fission product produced by nuclear reactors and nuclear weapons detonations. Cesium-137 is rapidly absorbed into the bloodstream and is distributed throughout the active tissues of the body. Metabolically, cesium-137 behaves as an analog of potassium and is distributed throughout the body. Its daughter, Barium-137m, an isomer, is an energetic beta and gamma radiation source and emits a 0.662-megaelectron volt gamma ray. Absorbed cesium-137 results in essentially whole-body irradiation (Amdur, Doull, and Klaassen 1991). The radioactive half-life of cesium-137 is 30 years. Its biological half-life in adults is 50 to 150 days, and in children is 44 days. The whole body is the critical organ for cesium-137 exposure.

7.1.3.3 *Mercury.*

The chemistry of mercury in the environment is complex. It has various oxidation states, biotic and abiotic methylation and demethylation processes, complexation with organic and inorganic ligands, and differential solubility and volatility forms. Speciation is a major determinant of the fate, bioavailability, absorption, and toxicologic characteristics of mercury compounds.

Although the generally more toxic organic forms of mercury, such as methylmercury, are unlikely to persist in the environment, they may form in biotic tissues and are known to biomagnify through ecosystems, particularly aquatic systems (Wren 1986, Scheuhammer 1987).

Because of its chemical stability and lipophilicity, methylmercury readily penetrates the blood-brain barrier. Thus, the central nervous system is a major target organ in both mammals and birds. However, adverse reproductive effects have been reported. Methylmercury can be converted to inorganic mercury in muscle tissues. The homolytic cleavage of the mercury-carbon bond leads to generation of reactive intermediates, e.g., methyl and metal radicals, which cause cellular damage (Wren 1986; Scheuhammer 1987; Manzo et al., 1992). The inhalation "no observed adverse effects level" (NOAEL) and "lowest observed adverse effects level" (LOAEL) are 0.000 and 0.009 mg/m^3, respectively (EPA 1997a).

7.1.4 Risk Characterization

The characterization of risk involves combining results of the toxicity and exposure assessments to estimate health risks. These estimates are either a comparison of exposure levels with appropriate toxicity criteria for noncarcinogens or an estimate of the lifetime cancer risk associated with a particular intake for carcinogens. The nature and weight of evidence supporting the risk estimate and the magnitude of uncertainty surrounding the estimate are also considered in risk assessment.

To determine human health risks, contaminant intakes are compared to the applicable contaminant toxicity data. The complete results of BRA risk characterization process, including risk estimates for each of the retained sites, are presented in Appendix D of the RI/FS report (DOE-ID 1999a). The generalized equations for calculating carcinogenic risk and noncarcinogenic hazard quotients from contaminant intake and toxicity information are provided in the following subsections.

7.1.4.1   Carcinogenic Health Effects. The following equations are used to obtain numerical estimates, (i.e., unitless probability) of lifetime cancer risks. The risk probability is the product of intake and slope factor, as follows, in Equation (7-1):

$$Risk = Intake \times SF \tag{7-1}$$

where

Risk = potential lifetime cancer risk (unitless)

Intake = chemical intake (mg/kg/day), or radionuclide intake (pCi)

SF = slope factor, for chemicals $(mg/kg/day)^{-1}$, or radionuclides $(pCi)^{-1}$.

The linear low-dose equation shown in Equation (7-1) is valid at risk levels lower than 1E-02. In accordance with EPA guidance (EPA 1989), risks that are greater than 1E-02 are calculated using the following one-hit equation, Equation (7-2):

$$Risk = 1 - exp(-Intake \times SF) \tag{7-2}$$

where

Risk = potential lifetime cancer risk (unitless)

Intake = chemical intake (mg/kg/day), or radionuclide intake (pCi)

SF = slope factor: for chemicals $(mg/kg/day)^{-1}$ or radionuclides $(pCi)^{-1}$

To develop a total risk estimate for a given rate at a given site, cancer risks are summed across all potential carcinogens at the site as shown in Equation (7-3):

$$Risk_T = \sum Risk_i \tag{7-3}$$

where

$Risk_T$ = total cancer risk, expressed as a unitless probability for a given exposure and a given route

$Risk_i$ = risk estimate for the ith contaminant for the route.

Similarly, risk values for each exposure route are summed to obtain the total cancer risk for each site.

7.1.4.2   Noncarcinogenic Effects. Health risks associated with exposure to individual noncarcinogenic compounds are evaluated by calculating the hazard quotient (HQ). The HQ is the ratio of the intake rate to the reference dose, as shown in Equation (7-4):

$$HQ = Intake / RfD \tag{7-4}$$

where

HQ	=	noncarcinogenic hazard quotient (unitless)
$Intake$	=	chemical intake (mg/kg/day)
RfD	=	reference dose (mg/kg/day).

Hazard indices are calculated by summing hazard quotients for each chemical across all exposure routes. If the hazard index for any contaminant of potential concern (COPC) exceeds unity, potential health effects may be a concern from exposure to the contaminant of potential concern. The hazard index is calculated using Equation (7-5):

$$HI = \sum \frac{Intake_i}{RfD_i}$$ (7-5)

where

HI	=	hazard index for a given COPC (unitless)
$Intake_i$	=	exposure level (intake) for the ith COPC (mg/kg/day)
RfD_i	=	reference dose for the ith COPC (mg/kg/day).

In Equation (7-5), intake and reference doses are expressed in the same units and represent the same exposure time period. Hazard indices may be summed across multiple contaminants to develop a total hazard index for a site.

7.1.5 Qualitative Uncertainty Analysis

Risk assessment results depend on the methodologies applied to develop risk estimates. These analysis methods were developed over a period of several years by INEEL risk management and risk assessment professionals to provide realistic, yet conservative estimates of human health risks. Nonetheless, if different risk assessment methods had been used, the BRA would have likely produced different risk assessment results. To ensure the risk estimates are conservative (i.e., generate upper-bound risk estimates), health protective assumptions that tend to bound the plausible upper limits of human health risks were applied throughout the BRA. Therefore, risk estimates that may be calculated by other risk assessment methods are not likely to be significantly higher than estimates developed for the OU 4-13 RI/FS.

Uncertainty factors are present in all four stages of risk analysis (i.e., data collection and evaluation, exposure assessment, toxicity assessment, and risk characterization). Uncertainties associated with parameters used in the risk assessment are listed in Table 7-1. The conservative assumptions and uncertainties in risk estimates for the three sites identified for remediation are summarized in Table 7-2. Qualitative consideration of the collective impact of all the assumptions indicates that risks are more likely to be overestimated than underestimated.

Table 7-1. BRA human health assessment uncertainty factors.

Uncertainty factor	Effect of uncertainty	Comments and Assumptions
Source term assumptions	May overestimate risk	All contaminants are assumed to be completely available for transportation away from the source zone. In reality, some contaminants may be chemically or physically bound to the source zone and unavailable for transport.
Natural infiltration rate	May overestimate risk	A conservative value of 10 cm/year was used for this parameter.
Moisture content	May overestimate or underestimate risk	Soil moisture contents vary seasonally in the upper vadose zone and may be subject to measurement error.
Water table fluctuations	May slightly overestimate or underestimate risk	The average value used is expected to be representative of the depth over the 30-year exposure period.
Mass of contaminants in soils is estimated by assuming a uniform contamination concentration in the source zone	May overestimate or underestimate risk	There is a possibility that most of the mass of a contaminant at a site may exist in a hotspot that was not detected by sampling. If this condition existed, the mass of the contaminant used in the analysis might be underestimated. However, 95% UCLs or maximum detected contamination were used for all mass calculations, and these concentrations are assumed to exist at every point in each waste site; therefore, the mass of contaminants used in the analysis is probably overestimated.
Plug flow assumption in groundwater transport	Could overestimate or underestimate risk	Plug flow groundwater models will likely estimate a greater mass of contaminants will be transported to the aquifer than would occur under natural conditions, with respect to concentrations because dispersion is neglected, and mass fluxes from the source to the aquifer differ only by the time delay in the unsaturated zone (the magnitude of the flux remains unchanged). For nonradiological contaminants, the plug flow assumption is conservative because dispersion as completed in the models is not allowed to dilute the contaminant groundwater concentrations. For radionuclides, the plug flow assumption may or may not be conservative. Based on actual travel time, the radionuclide groundwater concentrations could be overestimated or underestimated because a longer travel time allows for more decay. If the concentration decreases because the travel time delay is larger than the neglected dilution from dispersion, the model will not be conservative.
All infiltration into WAG 4 is assumed to occur through the contaminated sites	Will overestimate risk	Infiltration that normally occurs between contaminated sites is assumed to be concentrated on contaminated sites. This assumption results on a probable overestimate of risk because more water is available in the model calculation to carry contaminants to the aquifer.
No migration of contaminants from the soil source prior to 1994	Could overestimate or underestimate risk	The effect of not modeling contaminant migration from the soil before 1994 is dependent on the contaminant half-life, radioactive in growth, and mobility characteristics.
Contaminant source terms assumed to be lognormally distributed	Could overestimate risk	If sampling data at a given site fits a normal distribution rather than a lognormal distribution, the 95% UCL of the near concentrations calculated for the site could be as much as 50% too high.

Table 7-1. (continued).

Uncertainty factor	Effect of uncertainty	Comments and Assumptions
Chemical form assumptions	Could overestimate or underestimate risk	In general, the methods and inputs used in contaminant migration calculations, including assumptions made about the chemical forms of contaminants, were chosen to err on the protective side. All contaminant concentration and mass are assumed available for transport. This assumption results in a probable overestimate of risk.
Exposure scenario assumptions	May overestimate risk	The likelihood of future scenarios has been qualitatively evaluated as follows: resident - improbable; industrial - credible. The likelihood of future on-INEEL residential development is small. If future residential use of this site does not occur, then the risk estimates calculated for future on-INEEL residents are likely to overestimate the true risk associated with future use of this site.
Exposure parameter assumptions	May overestimate risk	Assumptions regarding media intake, population characteristics, and exposure patterns may not characterize actual exposures.
Receptor locations	May overestimate risk	Groundwater ingestion risks are calculated for a point at the downgradient edge of an equivalent rectangular area. The groundwater risk at this point is assumed to be the risk from groundwater ingestion at every point within the WAG 4 boundaries. Changing the receptor location will affect only the risks calculated for the groundwater pathway because all other risks are site-specific or assumed constant at every point within the WAG 4 boundaries.
For the groundwater pathway analysis, all contaminants are assumed to be homogeneously distributed in a large mass of soil	May overestimate or underestimate risk	The total mass of each COPC is assumed to be homogeneously distributed in the soil volume beneath the WAG 4 retained sites. This assumption tends to maximize the estimated groundwater concentrations produced by the contaminant inventories because homogeneously distributed contaminants would not have to travel far to reach a groundwater well drilled anywhere within the WAG 4 boundary. However, groundwater concentrations may be underestimated for a large mass of contamination located in a small area with a groundwater well drilled directly downgradient.
The entire inventory of each contaminant is assumed to be available for transport along each pathway	May overestimate risk	Only a portion of each contaminant's inventory is actually transported by each pathway.
Exposure duration	May overestimate risk	The assumption that an individual will work or reside at a contaminated site for 25 or 30 years is conservative. Short-term exposures involve comparison to subchronic toxicity values, which are generally less restrictive than chronic values.
Noncontaminant-specific constants (not dependent on contaminant properties)	May overestimate risk	Conservative or upper limit values were used for all parameters incorporated into intake calculations.

Table 7-1. (continued).

Uncertainty factor	Effect of uncertainty	Comments and Assumptions
Exclusion of some hypothetical pathways from the exposure scenarios	May underestimate risk	Exposure pathways are considered for each scenario and eliminated only if the pathway is either incomplete or negligible compared to other evaluated pathways.
Poorly defined dermal absorption factor values for most WAG 4 contaminants	May underestimate risk	A lack of absorption factor values for most WAG 4 contaminants may mean that dermal absorption risks are higher than expected. The possibility of unacceptable dermal absorption from soil risks being produced by WAG 4 contaminants is considered to be unlikely.
Model does not consider biotic decay	May overestimate risk	Biotic decay would tend to reduce contamination over time.
Occupational intake value for inhalation	Slightly overestimates risk	Standard exposure factors for inhalation have the same value for occupational as for residential scenarios. The time of exposure is assumed to be the same in the risk calculations for occupational workers as it is for residents.
Use of cancer SFs	May overestimate risk	Nonradionuclide SFs are associated with upper 95th percentile confidence limits and radionuclide SFs are central estimates of cancer incidence per unit intake. They are considered unlikely to underestimate true risk.
Toxicity values are derived primarily from animal studies	May overestimate or underestimate risk	Extrapolation from animal to humans may induce error caused by differences in absorption, pharmacokinetics, target organs, enzymes, and population variability.
Toxicity values are derived primarily from high doses; most exposures are at low doses	May overestimate or underestimate risk	Assumes linearity at low doses. Tends to have conservative exposure assumptions.
Toxicity values and classification of carcinogens	May overestimate or underestimate risk	Not all values represent the same degree of certainty. All are subject to change as new evidence becomes available.
Lack of SFs	May underestimate risk	COPCs without SFs, may or may not be carcinogenic through the oral pathway.
Lack of RfDs	May underestimate risk	COPCs without RfDs may or may not have noncarcinogenic adverse effects.
Risk/HQs are combined across pathways	May overestimate risk	Not all of the COPC inventory will be available for exposure through all applicable exposure pathways.

Table 7-2. Summary of source-term uncertainties site with selected remedies.

ID No.	Release Sites	Source Term Uncertainties and/or Assumptions
CFA-04	Pond (CFA-674)	Exposure point concentrations used for depth interval and volume-weighted concentrations are based on the 95% UCL or maximum detected concentration, whichever is less, instead of average (arithmetic mean) concentrations. The area of contamination is assumed to exist uniformly across the site, even though only two of the six COPCs were detected in 100% of the site-wide samples. The other COPCs were detected in at least 48.0% of the samples. The area of contamination is assumed to exist uniformly across the site. Contamination is assumed to exist down to 5.5 m (18 ft), even though positive detections of chemicals in the vadose zone are reported only to a depth of 2.4 m (8 ft bgs). The depth of contamination is based on the assumption that mobility of dissolved phase chemicals in the vadose zone (i.e., waste water) at CFA-04 is 3 m (10 ft). This assumption is made to ensure that potential risks from exposures at CFA-04 are not underestimated (Section 8). These assumptions may cause the calculated risks at the site to be overestimated.
CFA-08	Sewage Plant Drainfield	*Exposure point concentrations used for depth interval and volume-weighted concentrations* are based on the 95% UCL or maximum detected concentration, whichever is less, instead of average (arithmetic mean) concentrations. Of the nine calculated site-specific exposure point concentrations, seven are based on the maximum detected concentration. The area of contamination is assumed to exist uniformly across the drainfield, even though site-wide detection frequencies for each of the three COPCs are no greater than 72.3%. Contamination is assumed to exist at 10 m (32 ft) bgs. The depth to basalt is assumed to occur at 10 m (32 ft). It is assumed that COPCs will not migrate downward beyond 10 m (32 ft) due to the presence of basalt at 10 m (32 ft). These assumptions may cause the calculated risks at the site to be overestimated.
CFA-10	Transformer Yard	Exposure point concentrations used for depth interval and volume-weighted concentrations are based on the 95% UCL or maximum detected concentration, whichever is less, instead of average (arithmetic mean) concentrations. The area of contamination is the area of the site based on process knowledge that there was no specific pattern of waste disposal. The maximum depth of contamination is 0.6 m (2 ft) bgs based on depths of measured concentrations. For purposes of evaluating residential exposure pathways, contamination from 0 to 3.05 m (0 to 10 ft) soil interval is assumed. This assumption is made to ensure that potential risks from exposures at CFA-10 are not underestimated (Section 8). These assumptions may cause the calculated risks at the site to be overestimated.

7.2 Ecological Risk Evaluation Summary

Results of the WAG 4 ecological risk assessment (ERA) will be integrated into an INEEL-wide evaluation of potential risks to ecological receptors as a component of the WAG 10 OU 10-04 ERA. The WAG 4 ERA was conducted as outlined in the guidance for the INEEL.

An ecological site and contaminant screening was conducted to determine which sites and contaminants would be subjected to further analysis in the comprehensive RI/FS. The screening was completed and documented as part of the OU 4-13 Work Plan (DOE-ID 1997b). A site-by-site evaluation of risks to ecological resources as a result of exposure to contaminants was developed in the RI/FS. The evaluation included a review of screening completed in the Work Plan to ensure that sites or contaminants were not inappropriately omitted from further evaluation. Complete details of the ERA are presented in Sections 7 and 8 of the OU 4-13 RI/FS report (DOE-ID 1999a). The primary components of the ERA, discussed below, include problem formulation, analysis, risk characterization, and transition to the INEEL-wide ERA.

7.2.1 Problem Formulation

The goal of the problem formulation step is to investigate interactions between the stressor characteristics (i.e., contaminant characteristics), the ecosystem potentially at risk, and potential ecological effects (EPA 1992b). Site screening was conducted to identify the sites that could pose unacceptable risk.

Contaminant screening and data evaluation were conducted to identify COPCs and define exposure point concentrations. For the most part, results of the data evaluation conducted for the human health BRA were applied to the ERA. For those contaminants that were not retained for evaluation in the human health risk assessment, additional data evaluation to support the completion of the ERA was performed. Contaminant concentrations were compared to background concentrations and ecologically based screening levels. All radioactive contaminants were eliminated on the basis of this comparison.

Site-specific data characterizing contaminant concentration in biota for the INEEL ERAs are sparse. Consequently, the definition of assessment and measurement endpoints (i.e., ecological receptors) is primarily based on pathway and exposure analyses. Pathway and exposure models for contaminated surface and subsurface media were combined with a food web analysis to characterize the potential risks illustrated in the complete ERA conceptual site model (see Figure 5-2).

7.2.2 Analysis

In the analysis component of the ERA, the likelihood and significance of an adverse reaction from exposure to stressors were evaluated. Exposure assessment involved relating contaminant migration to exposure pathways for ecological receptors. The behavior and fate of contaminants of potential concern in the terrestrial environment were presented in a general manner because formal fate and transport modeling was not conducted for the WAG ERA. The ecological effects assessment consisted of a hazard evaluation and a dose-response assessment. The hazard evaluation involved a comprehensive review of toxicity data for contaminants to identify the nature and severity of toxic properties. The doses from multiple media (surface and subsurface soil) identified at WAG 4 were developed and used to assess potential risk to receptors. Because dose-based toxicological criteria exist for few ecological receptors, it was necessary to develop appropriate toxicity reference values (TRVs) for contaminants and functional groups at INEEL. A semiquantitative analysis was used, augmented by qualitative information and professional judgment as necessary.

Exposures for each functional group, threatened or endangered species, and sensitive species were estimated based on site-specific life history and when possible, feeding habits. Quantification of group and individual exposures incorporated species-specific numerical exposure factors including body weight, ingestion rate, and the fraction of diet composed of vegetation or prey, and soil consumed from the affected area. Parameters used to model contaminant intakes by functional groups were derived from a combination of parameters that produced the most conservative overall exposure for the group. Parameter values and associated information sources are discussed in further detail in the RI/FS (DOE-ID 1999a). The development of TRVs for those contaminants targeted for remediation based on unacceptable ecological risks is described in the following subsections.

7.2.2.1 Lead.
Lead is a ubiquitous trace constituent in rocks, soil, plants, water, and air. The average concentration of lead in the earth's crust is 16 mg/kg (Eisler 1988). Lead has four stable isotopes with the following percentages of occurrence: Pb-204 (1.5%), Pb-206 (23.6%), Pb-207 (22.6%), and Pb-208 (52.3%). Lead occurs in four valence states: (1) elemental (Pb), (2) monovalent (Pb^+), (3) divalent (Pb^{+2}), and (4) tetravalent (Pb^{+4}). In nature, lead occurs mainly as Pb^{+2} and is oxidized to $Pb.^{+4}$ Metallic lead is relatively insoluble in hard water; some lead salts are somewhat soluble in water. Of the organoleads, tetraethyllead and tetramethyllead are the most stable and are highly soluble in many organic solvents but are fairly insoluble in water. Both undergo photochemical degradation in the atmosphere to elemental lead and free organic radicals. Organolead compounds are primarily anthropogenic (Eisler 1988).

Lead is neither essential nor beneficial to living organisms. Lead affects the kidneys, blood, bone, and the central nervous system. The effects of lead on the nervous system are both functional and structural. Lead toxicity varies widely with the form and dose of administered lead. In general, organolead compounds are more toxic than inorganic lead. A significant cause of mortality among regulatory waterfowl is ingestion of lead shot.

Hatchlings of chickens, quail, and pheasants are relatively tolerant to moderate lead exposure (Eisler 1988). Dietary levels of 500 mg/kg had no effect on hatchling growth of these species, and levels at 2,000 mg/kg of lead had no effect on survival (Hoffman et al. 1985 as cited in Eisler 1988). For avian herbivores, a TRV was estimated using a study of mallards (Dieter and Finley 1978). Altricial species are generally more sensitive to lead than precocial species (Eisler 1988) of avian insectivores. An oral study using European starlings (Osborn, Eney, and Bull 1983) was used to generate a TRV for trimethyllead chloride. Because organic lead compounds are generally more toxic than inorganic lead, the toxicity quotients generated using this TRV should be interpreted with caution. American kestrels (*Falco sparverius*) exposed to 50 mg/kg/day of metallic lead in diets exhibited no effects on survival or reproductive success (Colle et al. 1980). Using these studies, TRVs were developed for avian functional groups.

Studies of rats administered lead in drinking water (Kimmel et al. 1980), lead toxicity of calves (Zmudzki et al. 1983), and lead toxicity of dogs (DeMayo et al. 1982) were used to develop TRVs for mammalian receptors. A critical concentration of 2,000 mg/kg of lead in food on a dry weight basis for reproduction was reported in a study on the toxicity of lead nitrate to the isopod (*Porcellio scaber*).

The recommended screening benchmark concentration for phytotoxicity in soil for lead of 50 mg/kg was used as the TRV for terrestrial plants (Suter, Will, and Evans 1993).

7.2.2.2 Mercury.
Mercury exists in the environment in three oxidation states: the elemental state, +1 (mercurous) state, and +2 (mercuric) state. The factors that affect the predominant oxidation state in an environment are the oxidation-reduction potential and the pH of the system. Particle-bound mercury can be converted to insoluble mercury sulfide, which can be bioconverted into more soluble or volatile

forms that may reenter the atmosphere or be taken up by biota and bioaccumulated in the terrestrial food chain. Mercury forms many stable organic complexes that generally are more soluble in organic matter than in water. Inorganic and organic particles strongly sorb mercury. Mercury can be transformed in the environment by biotic and abiotic oxidation and reduction, bioconversion of organic and inorganic forms, and photolysis. Mercury can be strongly concentrated by living organisms (Callahan et al. 1979). The chemistry of mercury in the environment is complex, not only because of its various oxidation states, but also because of biotic and abiotic methylation and demethylation processes, complexation with organic and inorganic ligands, and the differential solubility and volatility of various forms. Because speciation is a major determinant of the fate, bioavailability, absorption, and toxicological characteristics of mercury compounds, lack of knowledge of the state of the mercury in INEEL soil is a large source of uncertainty in both exposure assessment and TRV development.

Though the generally more toxic organic forms of mercury are unlikely to persist in the environment, they (in particular, methylmercury) may be formed in biotic tissues and are known to biomagnify through ecosystems, particularly aquatic systems (Wren 1986; Scheuhammer 1987). Thus, to ensure that mercury TRVs for WAG ERAs are protective of receptors at all levels of ecological organization, TRVs are developed from studies of the toxic effects of organic mercury. This measure is highly conservative and tends to result in an overestimate of risks for receptors lower in the food web because the majority of mercury in soil and plants (i.e., the majority of exposures to plants and soil-dwelling and herbivorous animals) is expected to be inorganic.

Because of its chemical stability and lipophilicity, methylmercury readily penetrates the blood-brain barrier. Therefore, the central nervous system is a major target organ in both mammals and birds. However, reproductive effects have been reported at even lower doses. Methylmercury can be converted to inorganic mercury in tissues. The homolytic cleavage of the mercury-carbon bond leads to generation of reactive intermediates (e.g., methyl and metal radicals, which cause cellular damage) (Wren 1986; Scheuhammer 1987; Manzo et al. 1992).

The effects of mercury on avian herbivores, insectivores, and carnivores were evaluated. For herbivores, the effects of organic mercury compounds on galliformes (e.g., domestic chickens, quail, and pheasants) have been investigated by several groups. However, no study was reviewed that identified a NOAEL. The LOAEL for relevant endpoints (i.e., reproductive success) of several similar studies was found in a study of the effects of mercury on birds (Fimreite 1979). Reduced egg production, shell thickness, and hatchability in pheasants that were fed seed, treated with organomercurial fungicide, were observed. This study was selected over others because of its use of a wild species and lower dose levels. A TRV was derived from this study.

Three goshawks were fed a diet of chickens that had eaten wheat dressed with an organomercurial fungicide (Borg et al. 1970). Their tissues contained 10 to 40 ppm of mercury, mostly as methylmercury. The hawks died after 30 to 47 days, and their total mercury intake was about 20 mg/bird.

Two studies examined the effects of subchronic methylmercury exposure on the reproductive competence of male and female rats (Khera and Tabacova 1973; Khera 1973). The NOAEL identified for both sexes was 0.25 mg/kg/day. Much less information is available about methylmercury toxicity to herbivores. In a study of acute methylmercury toxicity in mule deer (*Odocoileus hemionus*), 17.88 mg/kg was said to be the lethal dose of 50% of the exposed organisms (Eisler 1987). A number of studies have examined the effects of chronic methylmercury ingestion on carnivorous mammals, particularly house cats (e.g., Albanus et al. 1972; Charbonneau et al. 1976; Eaton, Secord, and Hewitt 1980) and mink (e.g., Aulerich, Ringer, and Iwamoto 1974; Wobeser, Neilson, and Schiefer 1976; Wren et al. 1987). The study of the chronic toxicity of house cats was considered superior to other available studies because of its long duration (two years), use of relatively large group sizes, detailed examination of endpoints,

identification of both no-effect and effect levels, and administration of mercury via both contaminated fish and addition to diet (Charbonneau et al. 1976).

A TRV of 0.3 mg/kg was assigned for mercury for terrestrial plants based on the toxicological benchmark (Suter, Will, and Evans 1993).

7.2.3   Risk Characterization

Risk characterization is the final step of the ERA process. The risk evaluation determines whether risk is indicated from the contaminant concentrations and the calculated dose for the INEEL functional groups, threatened or endangered species, and species of concern. The risk characterization considers the uncertainty inherent in the assessment. For a WAG ERA, the risk characterization step has two components: a description of estimation of risk, and a summary of results.

Risk is estimated by comparing the calculated dose to the TRV. If the dose from the contaminant does not exceed its TRV (i.e., if the HQ is less than 1.0 for nonradiological contaminants), adverse effects to ecological receptors from exposure to that contaminant are not expected and no further evaluation of that contaminant is required. Hence, the HQ is an indicator of potential risk. Hazard quotients are calculated using Equation (7-6):

$$HQ = \frac{Dose}{TRV} \tag{7-6}$$

where

HQ = hazard quotient (unitless)

$Dose$ = dose from all media (mg/kg/day)

TRV = toxicity reference value (mg/kg/day).

HQs were derived for all contaminants, functional groups, threatened or endangered species, and species of concern identified in WAG 4 for each site of concern. When information is not available to derive a TRV, then an HQ cannot be developed for that particular contaminant and functional group or species combination.

An HQ greater than the threshold value indicates that exposure to a given contaminant, at the concentrations and for the duration and frequencies of exposure estimated in the exposure assessment, may cause adverse health effects in exposed populations. However, the level of concern associated with exposure may not increase linearly as the HQ values exceed the threshold value. Therefore, the HQs cannot be used to represent a probability or a percentage because an HQ of 10 does not necessarily indicate that adverse effects are 10 times more likely to occur than an HQ of 1. It is only possible to infer that the greater the HQ, the greater the concern about potential adverse effects to ecological receptors.

In general, the significance of a HQ exceeding 1 depends on: (a) the perceived "value" (i.e., ecological, social, or political) of the receptor (or species represented by that receptor), (b) the nature of the endpoint measured, and (c) the degree of uncertainty associated with the process as a whole. Therefore, the decision to take no further action, order corrective action, or perform additional assessment must be determined on a site-, chemical-, and species-specific basis. With the exception of threatened or endangered species (EPA 1992b), the unit of concern in ERA is usually the population as opposed to the

individual. Therefore, exceeding conservative screening criteria does not necessarily mean that significant adverse effects to populations of receptors are likely.

Three sites, CFA-04, CFA-08, and CFA-10, with ecological HQs up to 30,000, 30, and 5,000 respectively, were retained for evaluation of remedial alternatives in the Comprehensive Feasibility Study (DOE-ID 1999a). These sites also pose an unacceptable risk to human health. Six other sites will be evaluated for ecological risk as part of the WAG 10 Sitewide assessment. These sites are CFA-01, CFA-02, CFA-05, CFA-13, CFA-41, and CFA-43.

Principal sources of uncertainty apply to the use of data not specifically collected for ERA and in the development of exposure assessment. Uncertainties inherent in exposure assessment are associated with estimated receptor ingestion rates, selected acceptable HQs, estimated site usage, and estimated risk assessment parameters (e.g., plant uptake factors and bioaccumulation factors). Additional uncertainties are associated with the depicted site characteristics, the determined nature and extent of contamination, and the derived TRVs. A large area of uncertainty is the inability to evaluate risk to many receptors because of the lack of appropriate toxicity data for many chemicals. This is especially a problem for certain receptors such as reptiles. In addition, because of the conservative nature of assumptions made to compensate for the lack of site-specific uptake and bioaccumulation factors, ecologically based screening levels for some chemicals are lower than their sample quantitation and detection limits. In WAG-4 analysis, this occurs for metals, polychlorinated biphenyls (PCBs), and some other organics. All of these uncertainties likely influence risk estimates. Major sources and effects of uncertainties in the ERA are reviewed in Table 7-3.

7.2.4   Transition to the INEEL-Wide Ecological Risk Assessment

The third phase of the ERA process is WAG 10 (OU 10-04) ERA, which will integrate WAG ERAs to evaluate risk to the INEEL-wide ecological resources. This assessment will evaluate effects resulting from past contamination, and their potential for adversely impacting the INEEL-wide ecological resources including residual impacts from completed remedial actions.

Sites identified in the WAG 4 ERA with an HQ greater than 10, and a concentration greater than 10 times the background concentration, will be considered in the INEEL-wide ecological risk assessment. The INEEL-wide ERA will be conducted as a component of the comprehensive RI/FS for OU 10-04. The WAG 10 comprehensive investigation will be referenced during the five-year review process for WAG 4 to determine if the decisions implemented by WAG 4 are still protective of the environment. If the OU 10-04 ERA determines that those WAG 4 sites screened at greater than 10 times background, or HQ greater than 10, require further action, it will be determined during the WAG 4, five-year review. Future remediation may be necessary if the WAG 10 INEEL-wide assessment indicates that a cumulative ecological risk is exceeded for a population of receptors or if land-use changes.

7.3   Risk Assessment Summary

The human health and ERA results are summarized in Table 7-4. The risks and HQ for the three sites and their COCs selected for remedial action are shown.

At the CFA-04 Pond, risk assessment calculations indicate that mercury poses a potential unacceptable risk to future residential receptors via ingestion of homegrown produce. The calculated hazard index for this exposure route is 80. Cancer risk at CFA-04 was less than 1E-04. Mercury was detected at depths to 0.6 m (2 ft) below pond bottom. Mercury also poses an ecological risk at CFA-04.

Table 7-3. Sources and effects of uncertainties in the ecological risk assessment.

Uncertainty Factor	Effect of Uncertainty (Level of Magnitude)	Comment
Estimation of ingestion rates (soil and food)	May overestimate or underestimate risk (moderate)	Few intake (ingestion) estimates used for terrestrial receptors are based on data in the scientific literature (preferably site-specific) when available. Food ingestion rates are calculated by using allometric equations available in literature (Nagy 1987). Soil ingestion values are generally from (Beyer et al. 1994).
Estimation of bioaccumulation and plant uptake factors	May overestimate or underestimate risk and the magnitude of error cannot be quantified (high)	Few bioaccumulation factors or Plant Uptake Factors are available in the literature because they must be both contaminant- and receptor-specific. In the absence of more specific information, Plant Uptake Factors and bioaccumulation factors for metals and elements are obtained from (Baes et al. 1994), and for organic compounds from (Travis and Arms 1988).
Use of human health exposure concentrations	May overestimate (high) risk	Exposure concentrations were derived from data obtained as a product of biased sampling of WAG 4 sites. Samples were generally obtained from areas where contamination was believed the greatest.
Estimation of toxicity reference values	May overestimate (high) or underestimate (moderate) risk	To compensate for potential uncertainties in the exposure assessment, various adjustment factors are incorporated to extrapolate toxicity from the test organism to other species.
Use of functional grouping	May overestimate (high) risk	Functional groups were designed as an assessment tool that would ensure that the ERA would address all species potentially present at the facility. A hypothetical species is developed using input values to the exposure assessment that represents the greatest exposure of the combined functional group members.
Site use factor	May overestimate (high) or underestimate (moderate) risk	Site use factor is a percentage of the site of concern compared to the home range. This is extrapolated from literature values and allometric equations and may vary from season to season and year to year depending on environmental conditions. It is highly uncertain.

Table 7-4. Summary of major risks and hazard quotients at individual sites and contaminants of concern that are addressed by the selected remedy for WAG 4.

Site	COC	Exposure Pathway	Risk	Hazard Quotient
Future Residential Exposure Scenario				
CFA-04	Mercury	Ingestion of homegrown produce	b	80
CFA-08	Cesium-137	External radiation exposure	4E-04	NA[d]
CFA-10	Lead	Ingestion of soil	a	a
Current Occupational Scenario				
CFA-04	Mercury	Ingestion of soil	b	0.3
CFA-08	Cesium-137	External radiation exposure	2E-03	NA[d]
CFA-10	Lead	Ingestion of soil	a	a
Future Occupational Scenario				
CFA-04	Mercury	Ingestion of soil	b	0.3
CFA-08	Cesium-137	External radiation exposure	2E-04	NA[d]
CFA-10	Lead	Ingestion of soil	a	a
Ecological Risk Assessment				
CFA-04	Mercury	Ecological exposure		<1 to 30,000
CFA-10	Lead	Ecological exposure		<1 to5,000
CFA-10	Copper	Ecological exposure		<1 to 70[c]

a. Risks and hazard quotients could not be estimated for lead because human health toxicity data are not available. However, concentrations in excess of the EPA screening level of 400 mg/kg (EPA 1994b) will be remediated.
b. Risk is less than 1E-04.
c. Copper contamination exists in the surface soil and any remedial action for lead contamination is expected to also remove copper.
d. NA–Not Applicable.

The carcinogenic risks at the CFA-08 Drainfield are greater than 1E-04 for external radiation exposure to current and future occupational workers and future residents to cesium-137. The noncarcinogenic HI at CFA-08 is less than one. Cesium-137 was detected from ground surface to between 1.2 m (4 ft) and 2.4 m (8 ft) bgs. Concentrations of cesium-137 are highest in the top 0.9 m (3 ft) of soil.

Lead was detected in surface soil between 0 to 0.6 m (0 to 2 ft) bgs at the CFA-10 Transformer Yard site. There are no toxicity data available for lead. Five samples reported concentrations above the 400 mg/kg EPA screening level. Lead also poses a risk to ecological receptors at CFA-10.

Groundwater risks were evaluated for 26 COCs identified in the OU 4-13 RI/FS (DOE-ID 1999a). The GWSCREEN modeling results indicate that WAG 4 does not contain sources of contamination that have the potential to produce risk greater than 1E-04 or an HQ greater than 1 for those COCs via the groundwater pathways (e.g., groundwater ingestion). No collection of sites showed risks in the air and groundwater residential scenarios greater than threshold values.